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LABORATORY SEDIMENT
BIOASSAY REPORT
ON LYONS CREEK SEDIMENTS
1992 & 1996

APRIL 1998

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LABORATORY SEDIMENT
BIOASSAY REPORT
ON LYONS CREEK SEDIMENTS
1992 & 1996

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EXECUTIVE SUMMARY

Sediments in Lyons Creek were sampled in the Spring of 1992 and the Spring of 1996, as part of an assessment in characterizing the potential ecological impacts of contaminated sediments for this area. In 1992, test sediment was sampled from five sites extending from the Welland Canal and eastward for a distance of one kilometre, within the city of Welland, Ontario. Two of the sites were resampled in 1996.

An objective of this study was to assess the spatial pattern of sediment toxicity and chemical bioaccumulation using static, laboratory sediment toxicity tests. Three independent toxicity tests were performed on whole-sediment samples. Mortality, growth and avoidance behaviour of the burrowing mayfly, *Hexagenia limbata*, was measured in 21-day exposures. Chironomid (*Chironomus tentans*) growth and mortality was determined in 10-day tests. Mortality and chemical uptake by the fathead minnow, *Pimephales promelas*, was examined by a standard 21-day test.

The 1992 sediment toxicity test results indicated sediment from two locations (station T5 and station 4) to be toxic to both benthic invertebrates, resulting in at least 40% mortality. For the remaining test exposures, station 5 and station T3 sediment yielded poor midge and mayfly growth, relative to the reference control sediment. Analysis of the chemical data indicated sediment polychlorinated biphenyl concentrations in the test sediments of 1.0 μg/g to 6.0 μg/g or 27 μg/g to 113 μg/g OC could explain the sublethal data but did not account for the lethal effects. A possible explanation is the co-occurrence of other organic chemical(s) those sediments with a total PCB sediment concentration above 3.0 μg/g (> 62 μg/g OC).

Sites that were sampled and tested in both 1992 and 1996 differed in PCB sediment contamination, as well as the level of biological effect. These differences could be a result of site heterogeneity and the distribution of contaminants within the creek over time, in addition to sampling depth and sample handling. Unfortunately, no conclusive statement can be made regarding other locations due to the small data set.

A commonality between the 1992 and 1996 data sets was the high availability of total PCBs to the fathead minnow, *Pimephales promelas*. The whole-body tissue concentrations measured after the 21-day exposure period consistently surpassed the International Joint Commission Aquatic Life PCB guideline of 100 ng/g (wet weight). The body residue concentrations of 630 ng/g to 2,480 ng/g also fell within the sportsfish consumption restriction guidelines. A significant relationship was found between sediment total PCB concentration and the fish tissue concentration. The ratio of fish to sediment PCB concentrations indicated substantial uptake not unlike those achieved through the ingestion of contaminated material.

In summary, the Lyons Creek test sediments received variable sediment rankings ranging from slight, moderate and low sediment quality depending upon the number of significant test responses and the degree of PCB uptake. An important finding from the minnow sediment test was the high availability of PCBs to fish from the test sediments. The whole-organism PCB concentrations and the subsequent biota-sediment accumulation factors indicated the surficial sediment are acting as a source of PCBs and appear to confirm levels achieved *in-situ.*

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1.0 INTRODUCTION

In 1991, West Central Region and Welland District Office of the Ontario Ministry of Environment (OMOE) began a study on Lyons Creek in an area near the Welland Canal after discovering a zone of polychlorinated biphenyl (PCB) sediment contamination. Additional field investigations were completed in the Fall of 1991 and Spring of 1992, with the assistance of former Water Resources Branch (WRB) staff. The purpose of these investigations was to determine the extent of sediment contamination and effects on benthic community structure, chemical bioaccumulation in caged mussels, *in-situ* forage fish, benthic invertebrates and sportsfish and laboratory toxicity as outlined in the report by OMOE (1998).

The initial studies found the contamination was historical in nature given the pattern of increasing PCB concentrations in the deeper sediment layers. The surficial sediment was considered a continuous source of PCBs and indeed there was evidence to suggest PCBs were being biomagnified through the food web with contaminated sediment acting as a source (OMOE, 1998). Some PCBs are known to be highly persistant and can reach elevated levels in sediment and biomagnify in animal tissue. Biomagnification is a term that describes the transfer of chemical from food to consumer, such that the residue concentration increases from one trophic level to the next (Rand and Petrocelli, 1985). It is for these reasons that PCBs appear on the OMOE primary list of banned substances (OMOE, 1992).

Based on the outcome of the 1992 study, additional work was carried out in July 1996 to ascertain the relative PCB distribution in creek sediments over the long term and to determine any change in sediment quality. Standards Development Branch (SDB) assisted in both studies by examining the association between inorganic and organic sediment contamination and biological effects using documented laboratory sediment toxicity test procedures (Bedard *et al.*, 1992). The data reported herein is meant to provide complementary site-specific information and, in conjunction with other biomonitoring data, may be used in future decision-making.

Whole-sediment toxicity tests were conducted from four field locations in 1992 and from two locations in 1996, using the mayfly nymph, *Hexagenia limbata* (21-day exposure, survival and growth), the midge larvae, *Chironomus tentans* (10-day exposure, survival and growth) and the juvenile fathead minnow, *Pimephales promelas* (21-day exposure, survival and chemical bioaccumulation). The battery of sediment toxicity tests provides several endpoints using organisms representing different trophic levels in order to measure differences in sediment quality. The laboratory toxicity tests are a cost-effective technique for determining whether sediment-associated contaminants are harmful to benthic organisms or are being released into the water column. In conjunction with appropriate negative and reference control sediments, spatial differences in sediment quality, the relative availability of contaminants and their potential impacts can be ascertained. Sediment contaminant concentrations were based on samples prepared for laboratory toxicity testing. The sediment was analysed for particle size, nutrients, metals, total PCBs, organochlorine pesticides and chlorinated benzenes. Surviving fathead minnows were submitted for whole-body tissue analysis of total PCBs.

2.0 MATERIALS AND METHODS

2.1 Sample Collection and Site Description

In early June 1992, surficial sediment was collected at five locations along Lyons Creek extending from the Welland Canal to just east of Hwy 140. The sampling locations were designated by OMOE, West Central Region, Welland District Office and the former Water Resources Branch, for sediment chemistry, field benthic community analysis, laboratory sediment toxicity testing and *in-situ* tissue analysis of caged or indigenous fish species (OMOE, 1998; Table 1). In 1992, test sediment was collected either from previously designated transects (T3 and T5) or newly assigned stations including those near highway 140 (stn 4 and stn 5) (Figure 1).

Two locations that had high PCB sediment contamination in 1992 were resampled in late June 1996 for additional laboratory toxicity testing. The sites were situated along transect 3 (stn T3) and transect 5 (stn T5).

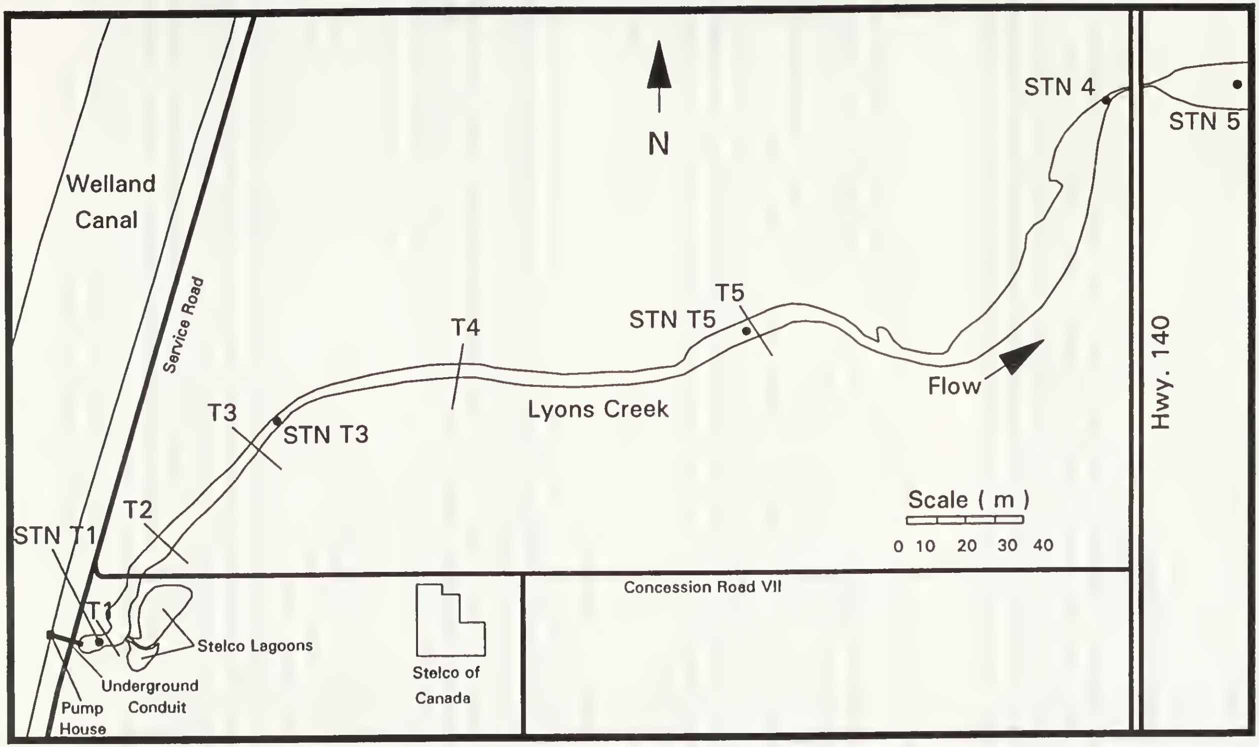
TABLE 1. 1992 and 1996 Lyons Creek Sampling Station Locations

Transect or Station Number	**Station Location**
T1	Reference control site; 1992; 23 m east of Welland Canal; 10 m west of Stelco outfalls
T3	Lyons Creek (East); 100 m downstream of stn T1
T5	Lyons Creek (East); 160 m west of Highway 140; opposite lot 144
4	Lyons Creek (East); 15 m west of Highway 140
5	Lyons Creek (East); 28 m east of Highway 140

Sampling was done using an Ekman grab sampler. At each station, approximately 15 L of composited surficial sediment (top 5 to 10 cm) was collected from several grabs. The composited sediment was placed into 20 L plastic buckets lined with food-grade polyethylene bags and transported to the Toronto, Ontario laboratory where they were stored at 4°C until required.

In both studies, a reference control sediment was sampled in a relatively, low-level contaminated area. In 1992, an upstream sample was collected near the pumping station along the east shore of the Welland Canal and southwest of the Stelco outfall (stn T1). For the 1996 study, sediment from a nearby creek was collected and it consisted primarily of silt and clay which supported bottom dwelling organisms including snails, chironomids (midges) and oligochaetes (worms). The reference control sediment should be representative of naturally occurring background contamination levels for the study area and be physically similar to the test sediments to discriminate effects due to physical or chemical causes. Sediment

Figure 1: 1992 and 1996 Lyons Creek Sediment Sampling Stations



collected from Balsam Lake or Honey Harbour, Ontario served as the negative control for the 1992 and 1996 bioassays, respectively. The negative control sediment is a relatively uncontaminated sediment that provides a measure of test acceptability (ASTM, 1997). Both control sediments are a basis for comparing the biological responses from the test sediments.

2.2 Analytical Methods

Chemical analysis of sediment and biota samples was conducted by the OMOE, Laboratory Services Branch, located in Toronto. Test methods are described in the *OMOE Handbook of Analytical Methods for Environmental Samples* (OMOE, 1983). Quality assurance procedures included method blanks, spikes, duplicates and standard reference materials. Analytical detection limits for the sediment and tissue parameters are listed in Table A1.

Sediment Nutrients and Particle Size Characterization

Homogenized bulk sediment (<2 mm fraction) was measured for total phosphorus (TP), total Kjeldahl nitrogen (TKN) and percent weight loss on ignition (LOI) which measured approximate organic content. Sediment total organic carbon (TOC) was determined with a LECO carbon analyzer using a dry combustion technique which oxidized carbon to CO_2. Particle size was measured on 50 g, air-dried samples using a Microtrac particle size analyzer for the size range 1.0 mm to 0.1 μm. This gives data on percent sand (2mm -62 μm), percent silt (62- 3.7 μm) and percent clay (3.7 - 0.1 μm) size classes. Detailed test methodology is described in OMOEE (1995a; 1995b).

Trace Metals in Sediment

Prepared sediment samples were digested using a concentrated aqua-regia acid mixture (1 part HNO_3 to 3 parts HCl). The dissolved trace metals including As, Cd, Cr, Cu, Fe, Pb, Mn, Ni and Zn in the digestates were detected by inductively coupled argon plasma atomic emission spectroscopy (ICP-AES), and Hg by flow injection vapour generated flameless atomic absorption spectroscopy (AAS). Detailed test methodology is described in OMOEE (1994a).

Organic Chemicals in Sediment

Moist sediment samples were solvent-extracted with acetone and dichloromethane. The extract was transferred to a rotary evaporator, concentrated and fractionated on a Florisil column. Different solvent combinations were used to elute the extracts into three groups: fraction A1 contained total PCBs, five Aroclor groups, hexachlorobenzene, heptachlor, aldrin, octachlorostyrene, pp-DDE and mirex; fraction A2 contained a- & b-BHC, a- & b-chlordane, op-DDT, pp-DDD, pp-DDT; and fraction A3 included heptachlor epoxide, oxychlordane, dieldrin, endosulfan I & II, endosulfan sulphate, endrin and methoxyclor. Analytes were identified and quantified using capillary gas chromatography equipped with a Ni^{63} electron capture detector (GLC-ECD). Detailed test methodology is described in OMOEE (1994b).

Organic Chemicals and Percent Lipid in Biota

Pooled whole fish samples (~5 g) were thawed, homogenized and acid digested using concentrated hydrochloric acid (HCl) on duplicate samples per station in both studies. The digestate was reacted with a mixture of 25% dichloromethane in hexane. The extract was treated with sodium bicarbonate to ensure neutralization and dried with anhydrous sodium sulphate. Dichloromethane-cyclohexane was added to the evaporated samples, followed by clean-up and detected by capillary gas chromatography equipped with a mass selective detector. Final results are reported on a wet weight basis for 14 chlorinated organic compounds and 14 pesticides. Percent lipid was determined on an aliquot (25 mL) of the final extract obtained before clean-up. The solvent was evaporated by air-drying in a fumehood for 24 hours and lipid residues were measured. Detailed test methodology is described in OMOE (1990).

2.3 Laboratory Biological Testing Methods

Basic Experimental Design

Sediment biological tests were conducted according to OMOE standardized procedures (Bedard *et al.*, 1992) and are briefly described below. The bioassays were static, single-species tests using whole-sediment. The experimental unit was a 1.8 L test chamber containing prepared sediment and dechlorinated municipal tap water (1:4, v:v). The chambers were randomly placed into a holding tank at ambient room temperature and maintained under a 16:8 hour, light:dark photoperiod and continuous aeration.

Moist field-collected bottom sediment was pressed through a 2-mm stainless-steel sieve to remove existing large biota and debris prior to use. Subsamples of this homogenized sediment were submitted for chemical and physical characterization according to standard OMOE procedures (OMOE, 1989). The sieved sediment was homogenized with a spatula and stored in 4 L acid-rinsed glass jars until required. Three hundred and twenty-five millilitre aliquots of homogenized sediment were placed into the test chamber and overlaid with the test water. After settling overnight, the chambers were aerated continuously until the end of the test. A clean, negative control sediment was collected from Balsam Lake (1992 study) or from Honey Harbour, Georgian Bay (1996 study). Negative control mortality must not exceed 15% for mayflies and fathead minnows and 25% for chironomids or the test is declared invalid.

Water in the exposure chambers was regularly monitored for pH, conductivity, total ammonia, un-ionized ammonia and dissolved oxygen. The 1992 exposures were not measured for ammonia. Dead organisms were removed and the numbers recorded daily. Any signs of abnormal behaviour of the test organisms or changes in appearance of the test chambers were noted. Water loss due to evaporation was replenished as needed.

Hexagenia limbata Lethality and Growth Assay

The 1992 toxicity test used three month old laboratory-reared mayfly nymphs with an average wet weight of 3.64 mg ± 0.32 (s.e.) (n=40). The 1996 toxicity test used five month old nymphs that weighed 3.47 mg ± 0.28 (s.e.) (n=36). The nymphs were raised from eggs collected by Dr. J. Ciborowski at the University of Windsor, Windsor, Ontario. Mayflies were reared according to OMOE procedures (Bedard *et al.*, 1992) and methods described in the literature (Friesen, 1981).

The rearing procedure involved the transfer of 600 newly-hatched nymphs to a 6.5 L aquarium which contained 2 cm of autoclaved sediment and 5.6 L dechlorinated tap water. Animals were maintained at ambient room temperature, 16:8 hour, light:dark photoperiod, continuous aeration and fed a vegetable diet.

Test organisms were retrieved from the rearing aquaria by sieving small portions of sediment in a 500-μm mesh brass sieve. The nymphs were washed into an enamel tray which held a fine mesh sieve and aerated, dechlorinated water. A Pasteur pipette (5-mm opening) was used to transfer the mayflies into 100 mL beakers of water and the contents were gently poured into the test chambers. Each test involved adding ten nymphs to each of the four replicate test chambers for a period of 21 days. Animals were not fed during the length of the test.

At the end of the test, the contents of each test chamber were emptied and rinsed in a sieve bucket. Surviving animals were counted and transferred to 150 mL beakers holding 100 mL dechlorinated water. The nymphs were immobilized with Alka-Seltzer®, blotted dry and individuals weighed to the nearest 0.01 mg.

Chironomus tentans Lethality and Growth Assay

Each toxicity test used 10-12 day old, cultured chironomid larvae weighing an average wet weight of less than 1 mg. The OMOE maintains continuous cultures of *C. tentans* larvae from egg to adult, following standard methods (Bedard *et al.*, 1992, Mosher *et al.*, 1982, Townsend *et al.*, 1981). Egg masses were originally supplied by Dr. J. Giesy at Michigan State University, Lansing, Michigan and have been cultured for several generations in our laboratory.

Initially, the midges were reared in enamel trays for 10 to 12 days and then maintained in a 21 L aquarium containing 1.6 L of silica sand. The cultures were held at ambient room temperature with continuous aeration and under a 16:8 hour, light:dark photoperiod. The larvae were provided a vegetable diet *ad libitum*.

Second and third instar larvae were directly transferred from the enamel rearing trays into the test chamber using the 5-mm opening of a Pasteur pipette. A total of 15 animals were added per chamber to each of the three replicates in the 1992 study and four test chambers were employed for the 1996 study. Animals were fed daily 30 mg of a Cerophyll®:Tetra Conditioning Vegetable® (3:2, w:w) diet.

After 10 days, the contents of the test chambers were emptied and washed in a sieve bucket. Surviving animals were sorted, removed and placed into 150 mL beakers holding 100 mL dechlorinated water and 15 mL silica sand. The larvae were counted, blotted dry and individuals weighed to the nearest 0.01 mg.

Pimephales promelas Lethality and Bioaccumulation Assay

The tests used cultured, juvenile fathead minnows with an average wet weight of 336 mg ± 26 (s.e.) (n=30) and 427 mg ± 26 (s.e.) (n=30), for the 1992 and 1996 toxicity tests, respectively. The minnows were cultured at the OMOE laboratory following techniques which for the most part are US EPA procedures (USEPA, 1987) with minor revisions (Bedard *et al.*, 1992).

Cultures were maintained at 20°C in a flow-through dechlorinated water system and under a 16:8 hour, light:dark photoperiod. Breeders were kept in 60 L glass aquaria and eggs were laid on spawning tiles. The tiles were incubated in a 25°C water-bath and the developing larvae were transferred to 400 L fibreglass holding tanks. Larval fish were fed 48-hour old live brine shrimp while juveniles and breeders were provided frozen brine shrimp. Each size class was fed *ad libitum*.

Each test chamber received 10 juvenile minnows for triplicate jars in 1996 and four replicates per sample for the 1992 study. The minnows were sorted into 250 mL glass beakers in groups of five. The contents of the beakers were emptied into a small net and the minnows released into the test chamber.

The minnows were exposed for 21 days and fed a Tetra Conditioning Vegetable® diet daily in an amount equivalent to 1% of the average starting wet weight. After 21 days the surviving fathead minnows were pooled from each replicate, counted, immobilized with Alka-Seltzer® and placed into 30 mL glass vials and frozen pending chemical analysis.

Reference Toxicant Testing

A water-only reference toxicity ($CuSO_4$) test was conducted with *H. limbata* and *C. tentans* for 48-hours and LC50s were calculated for the 1996 study only. The static tests consisted of four test concentrations and a control. The nominal copper concentrations were 0.05, 0.25, 0.5, 1.0 and 3.0 mg/L. Ten mayfly nymphs or midge larvae were placed into each of four replicate 250 mL beakers. To help reduce stress, five glass tubes were placed into the mayfly test beakers and a fine layer of silica sand was added to the midge test containers. Water quality parameters were recorded at 0 and 48 hours. The mayfly test used five month old laboratory-reared mayfly nymphs with an average wet weight of 4.6 mg ± 0.4 (s.e.). The midge larvae were 10-12 day post-hatch with an average wet weight < 1 mg in each set of tests.

Bioassay Schedule for Lyons Creek 1992 Sediment Samples

Test Organism	Species	Starting Date ('92)	Completion Date ('92)	Test Duration
Mayfly	*Hexagenia limbata*	June 24	July 15	21 days
Chironomid	*Chironomus tentans*	June 19	June 29	10 days
Minnow	*Pimephales promelas*	June 19	July 10	21 days

Bioassay Schedule for Lyons Creek 1996 Sediment Samples

Test Organism	Species	Starting Date ('96)	Completion Date ('96)	Test Duration
Mayfly	*Hexagenia limbata*	July 11	August 1	21 days
Chironomid	*Chironomus tentans*	July 16	July 26	10 days
Minnow	*Pimephales promelas*	July 11	August 1	21 days

2.4 Statistical Methods

Statistical analyses were performed using the SAS® software package (SAS, 1985). Comparisons were made among the test and control sediments using One-Way Analysis of Variance (ANOVA) and Tukey's studentized range test (HSD) and planned comparisons (Steel and Torrie, 1960). Dunnett's one-tailed *t*-test was used solely to compare mortality between the control and test sediments and the associated minimum significant difference (MSD) was described as a measure of test sensitivity. Analysis was made on arc-sine transformed mortality data. Homogeneity of variance across groups was tested using Bartlett's test. Coefficients of variation (C.V. %) were calculated for each endpoint as a measure of test precision. Spearman rank correlation analysis was used to investigate the correlation among the different biological endpoints for each species and sediment characteristics. Simple linear regression was used to measure the strength of the relationship between tissue and sediment chemical concentrations. LC50's (including the associated 95% confidence limits) were calculated using software developed by Stephan (1977) and were estimated by probit analysis. All contaminant residues were converted to a dry weight basis using a dry weight ratio of 0.15.

3.0 RESULTS

3.1 Water Quality Test Parameters

Conductivity, pH and dissolved oxygen parameters were periodically measured on the overlying water for each test species in each of the two studies and are summarized in Tables 2; 2A. Values are reported as mean ± standard deviation.

Similar pH water quality measurements were recorded among the test sites, regardless of test species or study. The average pH recorded among the test sediments for 1992 and 1996 was 8.0 and ranged from 7.5 to 8.3. Conductivity readings were also similar among sites and between studies. Average conductivity was 490 μmho/cm in 1992 ($n=6$) and 535 μmho/cm in 1996 ($n=12$). Water quality readings were quite similar between the negative control exposures, as well as between the reference control exposures, despite the difference in source. Dissolved oxygen within the test jars remained above the minimum acceptable level (>4 mg/l) throughout the test (OMOEE, 1994c). Test temperature was at 20°C for each bioassay.

The amount of total ammonia present in the overlying test water was measured during the 1996 study only. After correction for temperature and pH, the converted un-ionized ammonia readings were also recorded in Table 2A. Each of the test and reference sediments resulted in un-ionized ammonia above the PWQO of 0.02 mgNH_3/L in all three toxicity tests. The average value among the test sediments was 0.31 mgNH_3/L as compared to the reference control reading of 0.04 mgNH_3/L. Each test sediment consistently had higher amounts of TKN relative to the reference sediments and this may help to explain the trend in ammonia loadings.

3.2 Sediment Characterization

The following sections summarize the sediment physical and chemical parameters to aid in the interpretation of the biological toxicity results. Chemical analysis is based on the sediment prepared for toxicity testing and results may differ from those reported for any field samples collected concurrently. Any dissimilarities are likely due to *in-situ* chemical heterogeneity and/or sampling depth and sample handling.

Physical and Nutrient Properties

Sediments were characterized for % sand (2mm-62μm), % silt (62-3.7μm), % clay (3.7-0.1μm), % loss on ignition (LOI), total organic carbon (TOC), total phosphorus (TP) and total Kjeldahl nitrogen (TKN) (Tables 3; 3A).

Each of the test sediments from both studies was classified as being a silt clay loam (Millar *et al.*, 1965). Each test sediment also contained moderate amounts of organic matter ranging from 38 mg/g to 69 mg/g TOC and was generally black in color and soft textured. Other nutrient parameters including TP and TKN were comparable among sites with the exception of stn 4, where the TP concentration of 3.2 mg/g exceeded the PSQG-SEL concentration of 2.0 mg/g. The 1992 and 1996 reference control sediments closely matched

TABLE 2. Mean (± s.d.) water quality characteristics in 1992 sediment bioassays.

Test Organism: Mayfly (Hexagenia limbata)[a] Test Temperature: 20.8°C (0.3)

Station	*pH*	*D.O.* mg/L	*Conductivity* umho/cm
Control	8.05 (.21)	8.6 (0.1)	307 (18)
Reference (T1)	8.34 (.16)	8.6 (0.1)	477 (73)
Stn T3	8.23 (.11)	8.6 (0.1)	473 (62)
Stn T5	8.36 (.10)	8.6 (0.1)	571 (171)
Stn 4	8.04 (.45)	8.5 (0.1)	461 (120)
Stn 5	8.28 (.29)	8.6 (0.1)	488 (125)

Test Organism: Midge (Chironomus tentans)[a] Test Temperature: 20.7°C (0.3)

Station	*pH*	*D.O.* mg/L	*Conductivity* umho/cm
Control	7.83 (.16)	8.7 (0.1)	321 (12)
Reference (T1)	8.23 (.12)	8.7 (0.1)	462 (63)
Stn T3	8.18 (.08)	8.7 (0.1)	454 (47)
Stn T5	8.28 (.37)	8.7 (0.1)	512 (126)
Stn 4	8.13 (.12)	8.7 (0.1)	421 (38)
Stn 5	8.17 (.08)	8.7 (0.1)	445 (83)

Test Organism: Minnow (Pimephales promelas)[a] Test Temperature: 20.6°C (0.4)

Station	*pH*	*D.O.* mg/L	*Conductivity* umho/cm
Control	7.34 (.13)	8.8 (0.1)	327 (35)
Reference (T1)	7.97 (.17)	8.8 (0.1)	471 (64)
Stn T3	7.84 (.18)	8.8 (0.1)	527 (115)
Stn T5	7.90 (.14)	8.8 (0.1)	726 (282)
Stn 4	7.55 (.17)	8.8 (0.1)	776 (360)
Stn 5	7.68 (.11)	8.8 (0.1)	576 (180)

[a] Sample size N=3.

TABLE 2A. Mean (± s.d.) water quality characteristics in 1996 sediment bioassays.

Test Organism: Mayfly (Hexagenia limbata)[a] Test Temperature: 20.8°C (0.7)

Station	*pH*	*D.O.* mg/L	*Conductivity* umho/cm	*Total Ammonia* mg/L	*Un-ionized Ammonia* mg/L
Control	7.95 (.07)	8.3 (0.0)	297 (9)	0.17 (0.14)	0.006
Reference	8.39 (.05)	8.2 (0.1)	463 (64)	0.50 (0.79)	0.041
Stn T3	8.37 (.08)	8.2 (0.2)	461 (43)	1.95 (2.14)	0.160
Stn T5	8.27 (.03)	8.1 (0.3)	475 (104)	2.72 (3.22)	0.225

Test Organism: Midge (Chironomus tentans)[b] Test Temperature: 20.9°C (0.7)

Station	*pH*	*D.O.* mg/L	*Conductivity* umho/cm	*Total Ammonia* mg/L	*Un-ionized Ammonia* mg/L
Control	7.88 (.00)	8.1 (0.3)	315 (11)	0.17 (0.11)	0.006
Reference	8.10 (.20)	8.1 (0.1)	454 (51)	1.10 (0.88)	0.042
Stn T3	8.11 (.08)	8.1 (0.2)	450 (36)	5.33 (1.52)	0.262
Stn T5	8.09 (.08)	8.1 (0.3)	442 (44)	6.50 (1.50)	0.247

Test Organism: Minnow (Pimephales promelas)[a] Test Temperature: 20.8°C (0.7)

Station	*pH*	*D.O.* mg/L	*Conductivity* umho/cm	*Total Ammonia* mg/L	*Un-ionized Ammonia* mg/L
Control	7.06 (.65)	7.3 (0.5)	333 (37)	0.96 (0.50)	0.012
Reference	7.81 (.23)	6.5 (1.9)	483 (70)	1.92 (1.96)	0.057
Stn T3	7.85 (.12)	7.3 (0.6)	581 (149)	20.0 (11.2)	0.659
Stn T5	7.63 (.21)	6.3 (2.3)	534 (111)	20.4 (11.1)	0.311

a Sample size N=4; b Sample size N=3;

Underlining indicates un-ionized ammonia concentrations that exceed the PWQO of 0.02 mg/L

TABLE 3. Sediment physical and nutrient characteristics in control(s) and Lyons Creek 1992 sediment used in sediment bioassays.

Station	*% Sand* (2mm-62um)	*% Silt* (62-3.7um)	*% Clay* (3.7-0.1um)	*% LOI*	*TOC* mg/g	*TP* mg/g	*TKN* mg/g
Balsalm Lake Control	44.1	44.1	8.0	3.8	17	1.0	2.0
Reference Station T1	0.0	69.7	30.3	3.5	8	0.7	1.5
Lyons Creek Station T3	0.0	72.5	27.5	6.7	32	0.8	2.3
Lyons Creek Station T5	0.3	64.4	35.3	9.2	49	1.4	3.6
W. of Hwy 140 Station 4	1.7	61.9	36.4	10.0	53	3.2	2.7
E. of Hwy 140 Station 5	0.1	62.7	37.1	7.9	38	1.2	2.4
PSQG SEL Conc (mg/g dry weight)					100	2.0	4.8

Shading indicate sediment nutrient concentrations that exceed PSQG-SELs.

TABLE 3A. Sediment physical and nutrient characteristics in control(s) and Lyons Creek 1996 sediment used in sediment bioassays.

Station	*% Sand* (2mm-62um)	*% Silt* (62-3.7um)	*% Clay* (3.7-0.1um)	*% LOI*	*TOC* mg/g	*TP* mg/g	*TKN* mg/g
Georgian Bay Control	4.0	70.6	25.7	5.3	22	0.9	1.9
Reference Control	1.0 <W	68.2	31.8	6.0	28	0.8	1.7
Lyons Creek Station T3	9.0	57.0	34.0	9.8	56	1.3	3.2
Lyons Creek Station T5	1.0 <W	59.9	39.4	9.5	69	1.0	3.8
PSQG SEL Conc (mg/g dry weight)					100	2.0	4.8

<W - Less than detection limit.

Shading indicate sediment nutrient concentrations that exceed PSQG-SELs.

the test sediments in terms of particle size composition but were less nutrient-enriched. Between the two negative control sediments, sediment collected from Honey Harbour was a better representation of physical characteristics measured in the test sediments.

Trace Metal Sediment Concentrations

Bulk sediment samples were analysed for 11 trace metals (Tables 4; 4A). The sediment metal concentrations were compared to Severe Effect Level (SEL) and Lowest Effect Level (LEL) concentrations as outlined in the Provincial Sediment Quality Guidelines (PSQGs) (Persaud *et al.*, 1992). The SEL is defined as that chemical concentration in the sediment that could be detrimental to the majority of the macrobenthos and the LEL is the sediment contaminant concentration which can be tolerated by most benthic species.

Zinc sediment concentrations most frequently surpassed the PSQG-SEL concentration in 1992 and 1996 test sediments. One-half of the test sediments were either at the SEL concentration of 820 μg/g to as much as 3-fold higher (Zn: 2,500 μg/g). The only other exceedence occurred for Fe at stn 4. At least 8 of 10 trace metals were measured above LEL sediment concentrations. In comparison, the reference control sediments showed background sediment metal concentrations were generally at or below LEL concentrations.

Organic Chemical Sediment Concentrations

Concentrations of 17 organochlorine pesticides and 13 chlorinated organic compounds in the 1992 and 1996 test sediments were below the respective detection limits (Tables 5; 5A). Trace amounts of g-BHC (2-3 ng/g) and endosulphan I (4-6 ng/g) were found at stns T3 and T5, in the 1996 study. In addition, dieldrin (8 ng/g <T) was reported for stn T3 sediment in 1996.

All test sediments contained measurable amounts of total PCBs in 1992 and 1996 (Tables 5; 5A). Sediment PCB concentrations were above the LEL concentration and remained below the SEL concentration (after correction for TOC). During sediment preparation, before setting up the sediment bioassays, it was noted that most of the test sediments had an oily odour. In addition, two of the test sediments had visible amounts of oil, which coincidentally also had the highest PCB sediment concentrations: stn T5 in 1992; 3,080 ng/g and stn 4 in 1992; 6,040 ng/g. At these sites, the total PCB sediment concentrations more closely approached their respective SEL concentration, and were within a 10-fold difference. In contrast, sediments with total PCB concentrations ranging from 600 ng/g to 2,400 ng/g contained no visible oil and were 12-fold to 46-fold lower than SEL concentrations.

3.3 Mayfly (*Hexagenia limbata*) 21-day Lethality and Growth Results

The biological data for the two endpoints, mortality and growth, are summarized in Tables 6; 6A, for 1992 and 1996, accordingly. Mortality in the control and reference sediment was nil in 1996 and <3% in 1992. In 1992, significantly higher mortality occurred for stn T5 (45% mortality) and stn 4 (40% mortality), relative to the other test and two control sediments (ANOVA; $p<0.0001$). Percent mortality ranged from 5.0% to 7.5% at stn T3 and

TABLE 4. Bulk concentrations of trace metals in control(s) and Lyons Creek 1992 sediment (µg/g dry weight) used in sediment bioassays.

Station	*Al %*	*As*	*Cd*	*Cr*	*Cu*	*Fe %*	*Hg*	*Mn*	*Ni*	*Pb*	*Zn*
Balsalm Lake Control	1.2	2.4	0.8 <T	25	12	2.2	0.03 <T	530	18	19	78
Reference Station T1	1.4	5.6	0.5 <T	22	22	1.7	0.02 <T	450	29	14	110
Lyons Creek Station T3	1.7	7.6	0.8 <T	34	40	2.3	0.05 <T	440	38	29	310
Lyons Creek Station T5	2.0	7.5	1.6	44	61	3.1	0.07	380	48	66	1400
W. of Hwy 140 Station 4	2.5	13.0	1.7	53	57	5.4	0.10	450	64	68	2500
E. of Hwy 140 Station 5	2.7	8.0	1.2	42	48	3.2	0.09 <T	380	52	43	750
PSQG SEL Conc.	NA	33	10	110	110	4.0	2.0	1100	75	250	820
PSQG LEL Conc.	NA	6.0	0.6	26	16	2.0	0.20	460	16	31	120

<T - Trace Amount; Shading indicate sediment trace metal concentrations that exceed PSQG-SELs.

Underlining indicate sediment trace metal concentrations that exceed PSQG-LELs; NA - Not Available.

TABLE 4A. Bulk concentrations of trace metals in control(s) and Lyons Creek 1996 sediment (µg/g dry weight) used in sediment bioassays.

Station	*Al %*	*As*	*Cd*	*Cr*	*Cu*	*Fe %*	*Hg*	*Mn*	*Ni*	*Pb*	*Zn*
Georgian Bay Control	1.4	4.1	0.3 <T	29	15	2.5	0.06	690	21	58	89
Reference Control	1.7	8.3	0.4 <T	22	16	2.6	0.03 <T	1200	32	16	94
Lyons Creek Station T3	1.8	9.1	0.9 <T	42	61	3.2	0.07	440	49	43	830
Lyons Creek Station T5	2.3	9.5	0.7 <T	39	47	3.1	0.07	330	50	36	480
PSQG SEL Conc.	NA	33	10	110	110	4.0	2.0	1100	75	250	820
PSQG LEL Conc.	NA	6.0	0.6	26	16	2.0	0.20	460	16	31	120

<T - Trace Amount; Shading indicate sediment trace metal concentrations that exceed PSQG-SELs.

Underlining indicate sediment trace metal concentrations that exceed PSQG-LELs; NA - Not Available.

TABLE 5. Bulk sediment concentrations for chlorinated organics and pesticides in reference control and Lyons Creek 1992 sediment (ng/g, dry weight) used in bioassays.

All Stations	Heptachlor	1 <W
	Aldrin	1 <W
	Mirex	5 <W
	a-BHC	1 <W
	b-BHC	1 <W
	g-BHC	1 <W
	a-Chlordane	2 <W
	g-Chlordane	2 <W
	Oxychlordane	2 <W
	op-DDT	5 <W
	pp-DDD	5 <W
	pp-DDT	5 <W
	pp-DDE	1 <W
	Methoxychlor	5 <W
	Heptachlor epoxide	1 <W
	Endosulphan I	2 <W
	Dieldrin	2 <W
	Endrin	4 <W
	Endosulphan II	4 <W
	Endosulphan sulphate	4 <W
	Hexachlorobutadiene	1 <W
	Octachlorostyrene	1 <W
	Hexachlorobenzene	1 <W
	123-Trichlorobenzene	2 <W
	124-Trichlorobenzene	2 <W
	135-Trichlorobenzene	2 <W
	1234-Tetrachlorobenzene	1 <W
	1235-Tetrachlorobenzene	1 <W
	1245-Tetrachlorobenzene	1 <W
	Hexachloroethane	1 <W
	Pentachlorobenzene	1 <W
	236-Trichlorotoluene	1 <W
	245-Trichlorotoluene	1 <W
Reference (Stn T1)	Total PCBs	100 <T
Station T3		600
Station T5		3080
Station 4		6040
Station 5		1040

<W - Not Detected; <T - Trace Amount.

Underlining indicate sediment PCB concentrations that exceed PSQG-LELs.

TABLE 5A. Bulk sediment concentrations for chlorinated organics and pesticides in reference control and Lyons Creek 1996 sediment (ng/g, dry weight) used in bioassays.

All Stations (exceptions listed below)	Heptachlor	1 <W
	Aldrin	1 <W
	Mirex	5 <W
	a-BHC	1 <W
	b-BHC	1 <W
	g-BHC	1 <W
	a-Chlordane	2 <W
	g-Chlordane	2 <W
	Oxychlordane	2 <W
	op-DDT	5 <W
	pp-DDD	5 <W
	pp-DDT	5 <W
	pp-DDE	1 <W
	Methoxychlor	5 <W
	Heptachlor epoxide	1 <W
	Endosulphan I	2 <W
	Dieldrin	2 <W
	Endrin	4 <W
	Endosulphan II	4 <W
	Endosulphan sulphate	4 <W
	Hexachlorobutadiene	1 <W
	Octachlorostyrene	1 <W
	Hexachlorobenzene	1 <W
	123-Trichlorobenzene	2 <W
	124-Trichlorobenzene	2 <W
	135-Trichlorobenzene	2 <W
	1234-Tetrachlorobenzene	1 <W
	1235-Tetrachlorobenzene	1 <W
	1245-Tetrachlorobenzene	1 <W
	Hexachloroethane	1 <W
	Pentachlorobenzene	1 <W
	236-Trichlorotoluene	1 <W
	245-Trichlorotoluene	1 <W
Station T3	g-BHC	2 <T
	Endosulphan I	4 <T
	Dieldrin	8 <T
Station T5	g-BHC	3 <T
	Endosulphan I	6 <T
Control	Total PCBs	20 <W
Reference		20 <W
Station T3		2400
Station T5		780

<W - Not Detected; <T - Trace Amount.

Underlining indicate sediment PCB concentrations that exceed PSQG-LELs.

TABLE 6. Summary of biological results on mayfly, midge and minnow sediment bioassays for control(s) and Lyons Creek 1992 sediments.

Mean values (± standard deviation) where sample size n=3 reps for midge and n=4 reps for mayfly and minnow tests.

Test Organism	*Hexagenia limbata* (Mayfly)		*Chironomus tentans* (Midge)		*Pimephales promelas* (Fathead Minnow)
Station	*% Mortality*	*Ave. Individual Body Weight* (mg wet wt.)	*% Mortality*	*Ave. Individual Body Weight* (mg wet wt.)	*% Mortality*
Balsalm Lake Control	A 2.5 (5)	B 5.46 (0.5)	AB 8.8 (7)	B 7.37 (1.0)	A 2.5 (5)
Reference Station T1	A 0 (0)	A 27.91 (4.2)	A 2.2 (3)	A 9.17 (0.6)	A 5.0 (6)
Lyons Creek Station T3	A 7.5 (5)	B 6.19 (0.9)	AB * 19.9 (13)	D 2.06 (0.3)	A 2.5 (5)
Lyons Creek Station T5	B ** 45.0 (20)	B 5.74 (1.4)	BC ** 40.0 (20)	D 1.21 (0.2)	A 7.5 (5)
W. of Hwy 140 Station 4	B ** 40.0 (14)	B 8.14 (1.1)	C ** 66.6 (11)	-	A 0 (0)
E. of Hwy 140 Station 5	A 5.0 (10)	B 8.73 (0.7)	A 2.2 (3)	C 3.88 (1.0)	A 0 (0)
% MSD	19.4	-	23.5	-	7.2
% C.V.	50.2	14.6	38.3	12.7	106
D.P.	4.5	13.3	4.4	15.3	2.3

** %Mortality value is significantly different than the control and reference sediment (Dunnett's 1-tailed t-test;$p<0.05$).

* %Mortality value is significantly different than the reference sediment only (Dunnett's one-tailed t-test;$p<0.05$).

A Means sharing a common letter within a column are not significantly different; Tukey's HSD test for % Mortality ($p<0.05$) and planned comparisons using LSMEANS for comparing Body Weight ($p<0.01$).

MSD - Minimum Significant Difference; C.V. - Coefficient of Variation; D.P. - Discriminatory Power.

TABLE 6A. Summary of biological results on mayfly, midge and minnow sediment bioassays for control(s) and Lyons Creek 1996 sediments.

Mean values (± standard deviation) where sample size n=3 reps for minnow and n=4 reps for mayfly and midge tests.

Test Organism	*Hexagenia limbata* (Mayfly)		*Chironomus tentans* (Midge)		*Pimephales promelas* (Fathead Minnow)
Station	*% Mortality*	*Ave. Individual Body Weight* (mg wet wt.)	*% Mortality*	*Ave. Individual Body Weight* (mg wet wt.)	*% Mortality*
Honey Harbour Control	A 0 (0)	C 4.13 (0.3)	A 3.3 (4)	A 14.38 (2.0)	A 0 (0)
Reference Control	A 0 (0)	B 10.16 (0.3)	A 3.3 (4)	A 13.66 (1.0)	A 0 (0)
Lyons Creek Station T3	A ** 7.5 (10)	C 4.89 (0.5)	A ** 16.6 (8)	B 3.86 (1.0)	A 10.0 (0)
Lyons Creek Station T5	A 0 (0)	A 12.68 (1.3)	A 6.6 (9)	A 13.27 (1.1)	A 0 (0)
% MSD	7.7	-	11.2	-	0
% C.V.	13.2	7.5	79.5	9.7	0
D.P.	2.2	11.1	1.9	9.8	-

** %Mortality value is significantly different than the control and reference sediment (Dunnett's 1-tailed t-test;$p<0.05$).

A Means sharing a common letter within a column are not significantly different; Tukey's HSD test for % Mortality ($p<0.05$) and planned comparisons using LSMEANS for comparing Body Weight ($p<0.01$).

MSD - Minimum Significant Difference; C.V. - Coefficient of Variation; D.P. - Discriminatory Power.

stn 5. All test sediments yielded smaller-sized mayflies relative to the reference control weights (ANOVA; $p<0.0001$). The average mayfly weight in the test sediment exposures was 7.2 mg, which represents a doubling of the starting weight of 3.6 mg. A doubling of weight is considered an average rate of growth, yet remained significantly lower than that attained under reference conditions (27.9 mg).

None of the 1996 test or control sediments were lethal to mayfly nymphs in 21-day exposures (ANOVA; $p<0.08$) and each treatment received a similar ranking using Tukey's multiple range test. Although the paired Dunnett's *t*-test found stn T3 percent mortality to be higher than either control mortality, the 7% mortality rate is below the acceptable mortality criterion of 15%, indicating little or no effect. Organism growth data was variable among sites. Station T5 sediment resulted in stimulated mayfly growth (12.6 mg), while stn T3 sediment resulted in impaired growth (4.8 mg). The latter value was near the initial test weight of 3.4 mg and was half the reference control weight. The fairly low rate of growth observed in either negative control sediments is likely attributed to the longer storage time (~12 months) versus the freshly collected reference control sediments (~ 3 weeks). Mayfly growth depends on the quality and quantity of detrital material found in the sediment, since supplemental feeding was not provided throughout the test.

3.4 Chironomid (*Chironomus tentans*) 10-day Lethality and Growth Results

Results for chironomid growth and lethality are reported in Tables 6, 6A. Chironomid mortality rate in both controls and for both studies, averaged 2% to 8%. This is well within the acceptable maximum control mortality of 25%. In 1992, significantly higher mortality relative to the control sediments occurred for stn T5 (40%) and stn 4 (66%) (ANOVA; $p<0.0002$). Mortality in the other sediments ranged from 2% to 19% and was not significantly different than the control and reference control sediment. Midge growth was consistently lower in each of the test sediments as compared to either control exposures (ANOVA; $p<0.0001$). On average, a severe 71% growth reduction was found at stns T3, T5 and 5. The poor growth exhibited at stn T5 was overridden by the moderate toxic effect.

No significant differences in midge mortality was found among the 1996 Lyons Creek sediments (ANOVA; $p<0.10$). Percent mortality ranged from 3% to 16% and was well below the maximum acceptable criterion of 25% mortality. There was a measurable difference in body weight at one site (ANOVA; $p<0.0001$). Station T3 had a significantly lower body weight (3.8 mg), relative to stn T5 and the control exposures (Ave: 13.7 mg). This translated into a 72% growth reduction.

3.5 Fathead Minnow (*Pimephales promelas*) 21-day Lethality Results

Juvenile fathead minnow percent mortality data is reported in Tables 6; 6A, for the 1992 and 1996 studies, respectively. In 1992, control survival was at least 95% for both the control and reference control exposures. Minnow mortality at all test locations was not statistically different either among sites (0%-7% mortality) or relative to the controls (ANOVA; $p<0.14$).

In the 1996 toxicity test, minnow survival remained at or above 90%. Although the level of toxicity at stn T3 (10% mortality) was significantly higher than control mortality (0% mortality), it is within the acceptable control criterion of 15% mortality.

3.6 Quality Assurance Data

An evaluation was made on the 1992 and 1996 data sets in order to determine the repeatability of the test results (Tables 6, 6A). Initially, the two studies were considered independent of each other and should be treated as such. However, upon closer examination the toxicity results showed a marked similarity in lethal and sublethal test response for the negative control and reference control sediments between 1992 and 1996. This high degree of consistency between the two data sets will allow for an examination of not only spatial but also temporal trends, albeit at a limited number of sites (stns T3 and T5).

The parameters used to measure data quality were consistent between the 1992 and 1996 toxicity tests. For example, the coefficient of variation was always better for the benthic sublethal data (Range: 7% to 14% C.V.). The same endpoints also had a correspondingly higher discriminatory power (Range: 9 to 15 D.P.). In other words, the growth data were less variable, provided more accurate test results and were more capable of showing differences among sites, as compared to the lethality data.

The mayfly and midge lethality response was effective in detecting small differences in lethality among sites, as reflected in the excellent minimum significant difference (MSD) values of 19% and 23%. The MSD is a quantitative measure of test precision which describes the ability to detect a significant effect in the paired response between the control versus the test sample. However, there was a higher degree of variability associated with the benthic lethality data which resulted in poor C.V. and D.P. Overall, among the three test species, the mayfly and midge were equally sensitive in measuring differences in sediment quality. Minnow mortality was the least sensitive test endpoint.

The 48-hour copper LC50 (95% C.I.) for the water-only reference toxicant exposures for *H. limbata* for the 1996 study was 0.39 (0.20 - 1.69) mg/L. This value was within the acceptable 48-h LC50 (± 2 s.d.) range of 1.40 (1.25) mg/L, according to a previous series of reference toxicant tests. Similarly, for *C. tentans*, the LC50 was 1.57 (1.24 - 2.13) mg/L, as compared to an expected 48-h LC50 (± 2 s.d.) of 1.35 (0.96) mg/L. This indicates that the relative sensitivity of the test organisms fell within a normal response range.

3.7 Chemical Bioaccumulation in *Pimephales promelas*

The examination of chemical availability to aquatic organisms is valuable for assessing the potential for chemical transfer through the food web. The primary objective of this test design is to make general observations on whole organism tissue concentrations as they relate to overall bulk chemical concentrations in the sediment and differences in chemical uptake among sites. Surviving fathead minnows were submitted for the analysis of total PCBs. All values are based on whole-body tissue concentrations. Values are provided as wet weight and converted using a dry to wet ratio of 0.15 in the calculation of biota-sediment accumulation factors or BSAFs.

The sources of organic compound accumulation to forage fish include direct contact with the sediment and uptake from the overlying water. Factors that control chemical accumulation by forage fish include those that affect chemical adsorption and desorption such as sediment organic content, particle size distribution and the chemical's solubility properties commonly expressed by the octanol-water partition coefficient, Kow (Lake *et al.*, 1990). Biotic factors affecting uptake include metabolism and lipid content (Boese *et al.*, 1995).

Table 7 reports the total PCB tissue concentrations (wet weight) for each replicate and as an average value ± standard deviations for each station. In addition, the corresponding sediment PCB concentration (dry weight) is reported. Measurable amounts of PCBs were detected in fathead minnows for each of the test sediments. In 1992, the highest PCB tissue concentration occurred at stn 4 (2,480 ng/g), followed by stn T5 (1,890 ng/g) and stn 5 (1,420 ng/g). The tissue concentrations were at least 7-times higher than the reference control minnow tissue concentration of 200 ng/g. The 1996 exposures resulted in PCB tissue concentrations of 660 and 1,400 ng/g in the test sediments, as compared to being non-detected in either batch of negative control and reference control minnows (20 ng/g).

Concentration factors were calculated to assess the relative availability of total PCBs for the test and reference control sediments. The biota-sediment accumulation factor (BSAF) is defined as the ratio of organic chemical concentration in the fathead minnow, normalized for percent lipid, to that in the bulk sediment after correction for organic content (Lake *et al.*, 1990, Ankley *et al.*, 1992). Each individual replicate that was analysed consisted of approximately 15 individual animals.

$$BSAF = (C_t / L) / (C_s / TOC)$$

where,

C_t = tissue PCB concentration (ng/g tissue, dry weight)
L = tissue lipid concentration (g/g tissue)
C_s = sediment PCB concentration (ng/g sediment, dry weight)
TOC = total organic carbon content of sediment (g/g sediment)

BSAFs > 1.0, indicating that the chemical found in the organism surpassed those levels found in the bulk sediment, were reported at each test location. Substantial PCB accumulation was noted in 1992 and 1996 for all test sediments. Minnow PCB concentrations were 14-times to 39-times greater than the PCB concentration measured in the sediment (after correction for TOC) in the 21-day laboratory tests. Fish data were based on unpurged animals and any sediment found in the gut would have been included in the whole-body tissue concentration. However, in order for the gut content to account for the entire tissue concentration, over 100% of fish weight would need to be comprised solely of gut material. Obviously, the majority of the reported tissue concentration is due to chemical accumulation within other fish tissues. Boese *et al.*, (1996) concluded that BSAFs corrected for gut content were not grossly different than lipid-corrected BSAFs and the former approach is a valid measure of bioaccumulation potential. In the reference control exposures, total PCB concentrations were either at trace levels or non-detected in both sediment and fish, indicating

TABLE 7. Total polychlorinated bipyhenyl concentrations in fathead minnows exposed to control(s) and Lyons Creek 1992 and 1996 sediments in the laboratory and associated biota-sediment accumulation factors.

Year	1992						1996			
Station	Control Balsalm Lake	Reference Stn T1	Stn T3	Stn T5	Stn 4	Stn 5	Control Georgian Bay	Reference Control	Stn T3	Stn T5
Sediment PCB (ng/g dry weight)	NA	100 <T	600	3080	6040	1040	NA	20 <W	2400	780
Minnow PCB (ng/g wet weight)	20 <W 20 <W	240 160 <T	660 600	1680 2100	3360 1600	1400 1440	20 <W 20 <W	20 <W 20 <W	1300 1500	880 440
Average ± s.d.	20 (0) <W	200 (56)	630 (42)	1890 (296)	2480 (1244)	1420 (28)	20 (0) <W	20 (0) <W	1400 (141)	660 (331)
BSAF (dry wt / dry wt)		13.4	7.0	4.1	2.7	9.1		6.7	3.9	5.6
BSAF (corrected) (tissue lipid/sediment OC)		7.9	28.1	22.3	14.5	26.7		12.5	18.2	39.1
BSAF (revised) (tissue lipid 5%)		2.1	4.5	4.0	2.9	6.9		3.7	4.3	7.8

<W - Not Detected; <T - Trace Amount Measured; NA - Not Analyzed; One-half the detection limit was used to calculate BSAFs.

no additional external sources of PCBs were found during the test.

The average lipid content of surviving minnows in 1992 was 1.0% ± 0.3 (s.d.; n=12) and 1.3% ± 0.2 (s.d; n=8) in 1996. The determination of percent lipid in biota can vary depending on the analytical method (Randall *et al.*, 1991). Therefore, the lipid content used in calculating the BSAF may be rather small and BSAFs were re-calculated using an average lipid content of 5%, which is commonly cited in the literature for juvenile fathead minnows (Nebeker *et al.*, 1989; McCarty *et al.*, 1992; van Wezel *et al.*, 1995). Even the more conservative BSAFs indicated a substantial degree of PCB availability for all test sites (Range: 2.9 to 7.8).

Regression analysis was carried out between total PCB concentration in fish and sediment for the combined 1992 and 1996 data sets (n=8). A significant positive relationship was found, where increasing total PCB concentrations in the sediment had a corresponding increasing total PCB concentration in fish ($p=0.0010$; $R^2=85$) (Figure 2). A similar linear response was found using PCB sediment concentrations normalized for TOC ($p=0.0008$; $R^2=86$). The difference in the maximum and minimum total PCB sediment concentrations among sites was 99%, as was the amount of TOC (89%).

4.0 DISCUSSION

Spatial and Temporal Trends in Sediment Toxicity and PCB Bioaccumulation

A ranking system was used to identify differences in sediment quality among sites for the two independent data sets. This was determined by the magnitude of an effect using statistical test methods and total PCB minnow concentrations as they relate to existing federal and provincial tissue guidelines. Each endpoint was considered as being either a significant, toxic (T) or non-significant, non-toxic (N) response. In addition, the lethality endpoint received a greater weighting over the respective sublethal endpoint, where applicable. The final rating is based on the total number of positive hits recorded for each of the five biological endpoints along with the PCB bioaccumulation data. Each sediment fell into one of the following classifications (listed from least impacted (high quality) to most impacted (very low quality)): non-impacted (high); slightly impacted (slight); intermediately impacted (moderate); strongly impacted (low); and very strongly impacted sites (very low) (Tables 8; 8A).

In 1992, two of the Lyons Creek test sediments received a low sediment quality ranking due to the level of lethality reported at stn T5 and stn 4. Percent mortality was at or above 40% in both the *Hexagenia* and *Chironomus* toxicity tests and total PCB sediment concentrations exceeded 3,080 ng/g. The minnow PCB concentration was also the highest reported (over 1,800 ng/g) and fell within the sportsfish consumption restriction guidelines of 500 ng/g to 4,000 ng/g (MOEE/MNR, 1997). Station 5 sediment was considered moderately impacted and stn T3 slightly impacted according to the degree of sublethal benthic effect. This was particularly apparent in the midge test where growth was 70% reduced. The above information coupled with minnow PCB concentrations greater than the International Joint Commission (IJC) Aquatic Life Guideline of 100 ng/g (IJC, 1988), also contributed to the overall ranking at these sites. Only the reference control sediment (stn T1) was deemed non-

FIGURE 2. Regression Analysis of Fathead Minnow 21-d total PCB Tissue Concentration on Bulk Sediment total PCB Concentration for 1992 and 1996

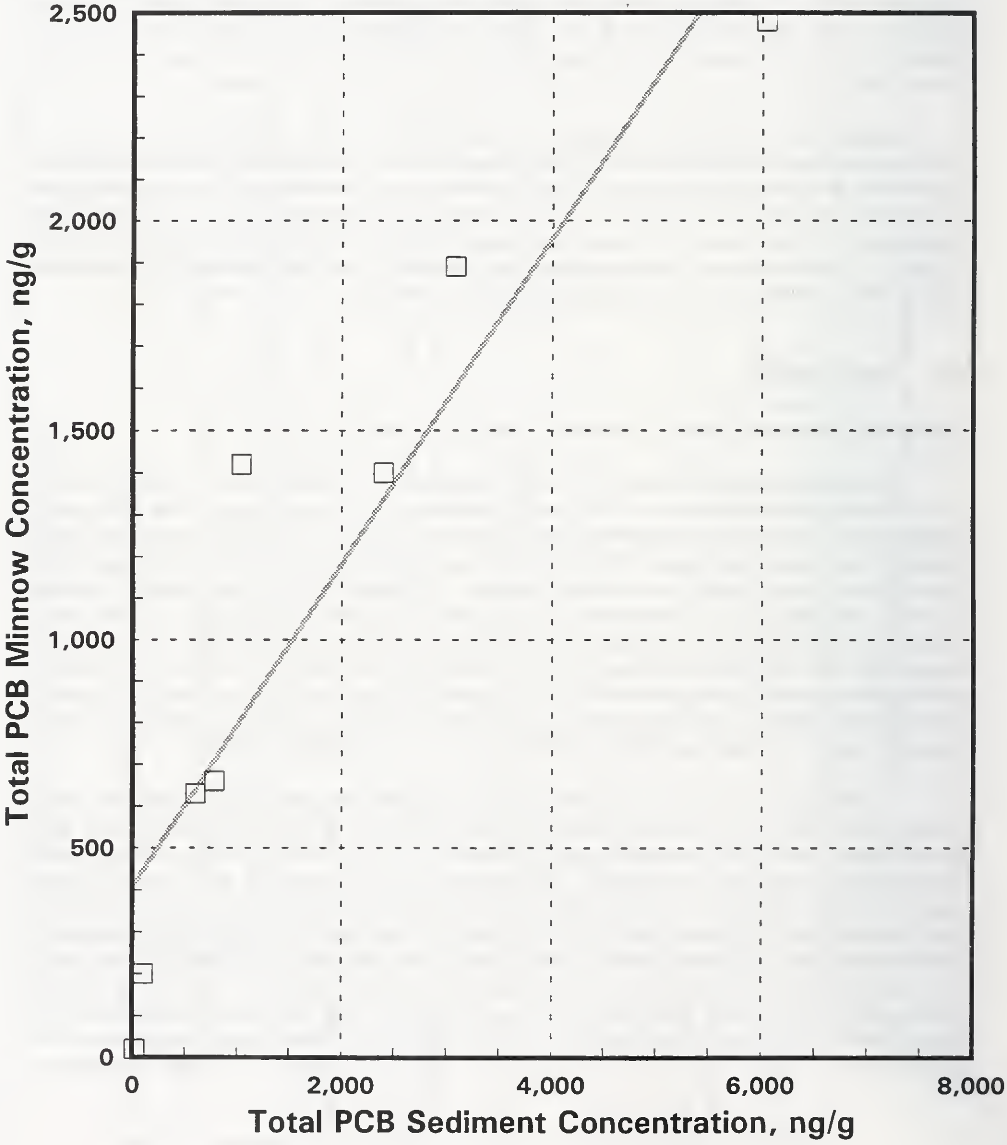


TABLE 8. Spatial variability in sediment toxicity and sediment quality for Lyons Creek 1992 samples.

Station	Sediment Quality	Sediment Total PCBs (ng/g dry wt)	Minnow Total PCBs (ng/g wet wt)	Mayfly Mortality	Mayfly Ave wt	Midge Mortality	Midge Ave wt	Minnow Mortality
Reference Stn T1	High	100 <T	200	N	N	N	N	N
Lyons Creek Stn T3	Slight	600	630	N	T	N	T	N
E. Hwy 140 Stn 5	Moderate	1040	1420	N	T	N	T	N
Lyons Creek Stn T5	Low	3080	1890	T	T	T	T	N
W.Hwy 140 Stn 4	Low	6040	2480	T	T	T	-	N

TABLE 8A. Spatial variability in sediment toxicity and sediment quality for Lyons Creek 1996 samples.

Station	Sediment Quality	Sediment Total PCBs (ng/g dry wt)	Minnow Total PCBs (ng/g wet wt)	Mayfly Mortality	Mayfly Ave wt	Midge Mortality	Midge Ave wt	Minnow Mortality
Reference Control	High	20 <W	20 <W	N	N	N	N	N
Lyons Creek Stn T5	Slight	780	660	N	N	N	N	N
Lyons Creek Stn T3	Moderate	2400	1400	N	T	N	T	N

N - Not Toxic, % mortality less than control criteria or $p>0.05$ and $p>0.10$ for growth data;

T - Toxic, % mortality greater than control criteria or $p<0.05$ and $p<0.10$ for growth data.

impacted.

Of the 1992 test sediments that were re-examined in 1996, neither had attained the same ranking. Station T3 had a poorer ranking in 1996, dropping from the classification of slight impairment in 1992 to being moderately impacted in 1996. Conversely, stn T5 went from a low ranking to being slightly impacted. At these sites there was about a 4-fold difference in sediment PCB concentration between the two sampling periods, either increasing or decreasing depending on the location. This was likely a result of *in-situ* heterogeneity, as illustrated in the PCB sediment distribution measured along a transect (OMOE, 1998) or could be a result of contaminant redistribution in certain portions of the creek over time. In terms of minnow PCB concentrations, both sites consistently resulted in tissue concentrations well above PCB tissue guidelines. As expected, the reference control sediment did not generate any adverse test response or quantifiable PCB uptake by fish.

The degree of test response observed in 1996 was very similar to those measured in 1992. In other words, sediment PCB concentration of 2,400 ng/g or greater elicited significant benthic growth impairment of 50% to 70%, while PCB sediment concentrations at or below 780 ng/g were not associated with a marked toxic effect.

Relationships Between Biological Endpoints and Sediment Physical/Chemical Characteristics

Spearman rank correlation coefficients were calculated among each test species and endpoint for the 1992 study only, owing to the suitable sample size ($n=5$) (Table 9). Among all the endpoints, benthic invertebrate survival and growth effects were highly intercorrelated ($r=+0.87$ to $+0.97$, $p<0.05$). Sites that had poor survival rates often had smaller-sized animals and helped to distinguish between non-toxic and toxic sediments. Generally fish mortality was an insensitive indicator of sediment quality. This lack of mortality suggests the fish had a higher tolerance towards the test sediments and should actually increase exposure to any sediment-associated or water-borne contaminants. Negligible fish avoidance increases the level of contact with the sediment when the fish are actively feeding near the sediment surface. For these reasons, the minnow test is expected to provide information describing relative PCB availability along a gradient of sediment contamination (Lester and McIntosh, 1994; Harkey *et al.*, 1994).

The toxicity endpoints were also compared to sediment physical, nutrient and chemical parameters to aid in determining potential causes for the observed laboratory effects in 1992 (Table 10). Sediment total PCB concentrations was a poor indicator of organism toxicity and was only slightly correlated with midge growth ($p<0.10$). The relationship between organism mortality and sediment PCB concentrations was not incremental but fell into two separate groups. The two sites (stn 4 and T5) in 1992 that had PCB sediment concentrations which most closely approached the PSQG-SEL, by 8-fold, resulted in significant toxic effects. The other group of sediments had PCB sediment concentrations that were 12-fold to 28-fold lower than the PSQG-SEL and had a non-significant degree of lethality. The lack of correlation between PCB sediment concentration and degree of biological effect implies other organic chemical(s), in addition to PCBs, contributed to the observed pattern of toxicity.

TABLE 9. Spearman rank correlation coefficients indicating significant positive (direct) correlations among toxicity data for Lyons Creek 1992 sediments.

	Mayfly Survival	Mayfly Growth	Midge Survival	Midge Growth
Mayfly Growth	+ .900 *			
Midge Survival	+ .872 *	n.s.		
Midge Growth	+ .900 *	n.s.	+ .974 **	
Minnow Survival	n.s.	n.s.	n.s.	n.s.

** $p < 0.05$; * $p < 0.10$; n.s. - Not Significant at $p>0.10$.

TABLE 10. Spearman rank correlation analysis summary indicating significant negative (inverse) or positive (correlation between biological endpoints and sediment physical and chemical parameters for Lyo Creek 1992 samples.

Toxicity Endpoint	Mayfly Survival	Mayfly Growth	Midge Survival	Midge Growth	Minnow Survival
Bulk Concentration	- Cu *			- TOC * - Cd * - Cr * - Pb * - Zn * - Total PCB *	
TOC corrected Concentration[a]	+ Cr ** + Cu * + Hg * + Ni *	+ Hg ** + Ni ** + Cr * + Fe *	+ Cu *	+ Cu ** + Cr * - Total PCB *	

[a] Included Cd, Cr, Cu, Fe, Hg, Ni, Pb, Zn and total PCBs.

** $p < 0.01$; * $p < 0.10$.

A literature review on sediment bioassays performed on PCB-contaminated sediments supports this idea. Swartz *et al.*, (1988), derived a 10-day LC50 for sediment spiked with Aroclor 1254 of 10.8 µg/g or 2,600 - 4,500 µg/g OC, using the saltwater amphipod, *Rhepoxynius abronius*. A similar lethal effects-concentration was reported by Murdoch *et al.*, (1997a) for the same species, with a 10-day LC50 of >27.4 µg/g or >2,560 µg/g OC. In a study using field-collected salt-water sediments, a no-effect concentration of >395 µg/g OC for total PCBs was determined using the marine polychaete, *Neanthes arenaceodentata* (Murdoch *et al.*, 1997b). In comparison, the highest PCB concentration in Lyons Creek was 6.0 µg/g or 113 µg/g OC.

Freshwater sediments collected in 1993 from the Detroit River having PCB sediment concentrations closely approximating those measured in Lyons Creek yielded similar test results. Besser *et al.,* (1996) found several sediments that caused growth reduction in the midge, *Chironomus tentans,* in excess of 50%. The corresponding PCB sediment concentrations were 1.2 to 6.1 µg/g or 30 µg/g OC to 121 µg/g OC. These values are 4-fold to 17-fold lower than the SEL and are similar to those reported for Lyons creek sediments. Wood *et al.*, (1997) performed a dilution series on PCB and PAH-contaminated field sediments in 12-day tests using the midge, *Chironomus tentans* and observed a 75% mortality rate at PCB concentration of 22.0 µg/g or 846 µg/g OC. In comparison, the highest PCB concentration in Lyons Creek was 6.0 µg/g or 113 µg/g OC. Overall, the PCB sediment concentrations can account for the sublethal effects but fail to adequately explain the level of lethality.

The presence of other co-occurring organic contaminants in the most lethal Lyons Creek sediments must be considered. A common feature observed in both stn T5 and stn 4 1992 sediments was the presence of not only an oily odor but also a visible oily sheen. A sheen was not noted in any of the other test sediments. Field studies have found the presence of visible oil in sediment to be a reliable marker in describing the relative abundance of benthic organisms, including mayflies (Hiltunen and Schloesser, 1983; Burt *et al.*, 1988; UGLCCS, 1989). Further chemical identification would help confirm whether other types of organic compounds are present.

Correlation analysis revealed trace metal sediment concentrations co-varied with PCB sediment concentrations. The strongest relationship occurred between total PCBs and Zn, Hg, and Fe, and to a lesser degree with Mn, Ni and Pb. Chemical analysis on field samples collected throughout Lyons Creek also showed this relationship, as if the contaminants were derived from a common source (OMOE, 1998). Among the metals, only Zn was measured above the PSQG-SEL (Max: 2,500 µg/g) at some sites. It is possible that elevated Zn, along with high PCB sediment concentrations, may be acting jointly towards organism toxicity. The question of joint chemical toxicity was examined in a laboratory sediment toxicity study which tested different groups of inorganic and organic chemicals, singly and in combination (Swartz *et al.*, 1988). The study found the additive effect of Zn and PCB (Aroclor 1254) was minimal. On the other hand, the same study found a 32% increase in amphipod mortality when more than one class of organic compound was tested (Aroclor 1254 and flouranthene). This further supports the probability that other organic compounds are acting as toxic agents rather than it being related solely to Zn.

Correlation analysis also found a negative relationship between midge growth and sediment TOC in the 1992 study (Table 10). The quantity of organic matter in the Lyons Creek sediments (>3% TOC) is considered suitable for midge survival, according to work by Suedel and Rodgers (1994). Midge larvae feed by grazing upon the food supplied during the test, thereby making growth less dependent on organic matter associated with the sediment (Ankley *et al.*, 1994). Interestingly, midge growth was the only endpoint that also correlated with sediment PCB concentration, which in turn co-varied with sediment TOC.

Another variable that may have influenced the outcome of the benthic toxicity tests is the relative number of extraneous oligochaetes found in most of the test sediments. Reynoldson *et al.*, (1994) demonstrated the ability of aquatic worms to negatively affect organism growth by as much as 90% for *Chironomus* and 50% for *Hexagenia*, depending on the worm density in the sample. Test sediments from Lyons Creek included either none, low, moderate or high numbers of oligochaetes, based on estimates made at the end of the midge test on sieved material caught on a fine-mesh screen. The reference control sediment used in the 1992 study contained a moderate number of oligochaetes and this did not adversely impact midge growth. Conversely, stn T5 in 1992 had few oligochaetes and still resulted in a severe midge growth reduction, indicating a potential adverse effect to both midges and oligochaetes. The divergent results make it difficult to assess what, if any, effect the presence of oligochaetes may have had on the test results.

Importance of PCB Bioavailability and Bioaccumulation

Substantial amounts of PCB were found to accumulate in whole fish in each of the test exposures, which raises a concern for its' transfer through the food chain via the sediment. Several studies have consistently found PCBs are more bioaccumulative relative to other types of organic chemicals, such as PAHs (Brannon *et al.*, 1993; Harkey *et al.*, 1994; Lake *et al.*, 1996; Tracey and Hansen, 1996; Kaag *et al.*, 1997; Wood *et al.*, 1997). Fathead minnows attained PCB residue concentrations in 21 days that were 2-times to 9-times those concentrations measured in the sediment. Even after normalizing for any variation in sediment TOC, the fish tissue concentrations were 14-times to 39-times higher than the respective total PCB sediment concentration, giving an average BSAF of 24.7 ± 8.6. Biota-sediment accumulation factors that are greater than 4.0 for highly bioaccumulative compounds e.g. PCBs, is used as a screening value in the risk assessment of dredged material (USEPA, 1991). This value was exceeded by most of the Lyons Creek sites tested in 1992 and 1996.

The percent lipid value used to calculate the latter BSAFs can differ depending on the analytical method employed, specifically, the type of extraction solvents used (Randall *et al.*, 1991). An underestimation of percent lipid will inflate the BSAF (Honeycutt *et al.*, 1995). Therefore, the ratios were re-calculated using a revised lipid content of 5%, based on literature values. The BSAFs (2.9 to 7.8) using the higher percent lipid still exceeded the screening value of 4.0. These revised BSAFs were not unlike those derived in other sediment exposures using mayflies, 5.0 to 9.2 (Drouillard *et al.*, 1996) and molluscs and polychaetes, 2.6 to 4.9 (Lake *et al.*, 1990). It should be noted that the BSAFs are based on 21-day exposures and may not necessarily reflect maximum PCB tissue concentrations that could be attained over a longer period of exposure. In a 28-day laboratory test using clams (*Macoma nasuta)*, final tissue concentrations were only 75% of predicted steady-state PCB concentrations (Boese *et al.*, 1995). Similarly, Drouillard *et al.*, (1996), estimated a 32-day exposure was necessary to approximate a steady-state condition.

Despite the obvious differences between laboratory and field conditions, the range in minnow PCB concentrations of 660 ng/g to 2,480 ng/g (wet weight) measured in this study, were quite comparable to those PCB tissue concentrations recorded for assorted species of fish collected in Lyons Creek. Hitchin (1997) observed elevated PCB levels in 1992 that ranged from 783 ng/g for rosyface shiners to 1,460 ng/g for spottail shiners. At a location near Hwy 140, in 1994; maximum PCB concentrations in fish flesh were 9,600 ng/g in carp and 3,380 ng/g in pumpkinseed (A. Hayton, MOEE, pers. comm.) and illustrated a high degree of biomagnification. A long-term biomonitoring study in Pottersburg Creek, Ontario also exhibited similar trends in PCB concentrations both in bottom-dwelling fish species (1,500 - 3,400 ng/g, wet weight) and in sediment (1,096 - 8,103 ng/g, dry weight) (Zaranko *et al.*, 1997).

Federal tissue guidelines provide some indication of the ecological relevance of PCB concentrations measured in fish. The PCB body burdens measured in the laboratory-exposed fathead minnows reached levels that consistently surpassed the IJC criterion of 100 ng/g (wet weight) that protects against the transfer of persistent substances through the aquatic food chain (IJC, 1988). Although the guidelines are based on removable portions of fish flesh, they have been applied to whole-organism young-of-the-year forage fish (Suns *et al.,* 1991).

Uptake of PCBs is often described as a simple partitioning of the chemical between the organic carbon fraction of sediment and the lipid portion of the animal. The association of TOC in controlling PCB availability has been frequently referred to and it is expected sediments with higher amounts of TOC will result in reduced availability due to chemical sorption (Lake *et al.,* 1990; Boese *et al*, 1995). Although there was a significant correlation between sediment TOC and PCB concentrations ($r=1.0$; $p=0.001$) among the test sediments, clearly PCB availability was independent of sediment TOC. Even with moderate amounts of sediment TOC (˜5.0%), PCB is readily available to fish, where one would anticipate lower uptake. Only at fairly high amounts of TOC (˜10%) does PCB uptake become compromised such as that observed for *in-situ* and laboratory organisms in a study investigating PCB availability in the Otonabee River (Jaagumagi and Petro, 1991). The ability of TOC to describe PCB availability in sediment to benthos has been questioned (Harkey *et al.*, 1994; Dabrowska and Fisher, 1993). Harkey *et al.*, (1994) suggests that binding to TOC may actually be less important than the influence of sediment particle size.

Partitioning alone can account for bioaccumulation factors that average 1.7 (Konnemann and Van Leeuwen, 1980; McFarland and Clarke, 1986). The higher BSAFs reported in this study cannot be explained solely due to simple partitioning but implies that an additional process is playing an important role. Increased chemical uptake can occur in organisms which directly ingest highly lipophilic chemicals in the form of food (prey or sediment). In our case, the sources of contaminants to fathead minnows could include the ingestion of sediment particles, suspended particulates and contaminated oligochaete worms that may have been present (Dabrowska *et al.*, 1996). Multiple routes of uptake are known to contribute to higher BSAFs (Lee, 1992). During ingestion, the food is partially digested and either the organic content or the lipid portion of the contaminated matrix is absorbed through the gut. This process further augments the assimilation of the chemical due to changes in chemical fugacity which eventually leads to substantial chemical magnification (Gobas *et al.*, 1993). This process has been confirmed both for predator/prey interactions (Russell *et al.*, 1995) and benthos/sediment interactions (Boese *et al.*, 1996). The BSAFs reported herein (2.9-7.8) agree favourably with those derived using the fugacity model (3.5-6.1) (Gobas *et al.*, 1989).

The PCB sediment and fish data are reported for total PCBs rather than congener-specific concentrations. This limits any interpretation of the test results with respect to which of the PCB congeners are more available or more toxic (Schweitzer *et al.*, 1997). Typically those PCB congeners found at the higher concentration in the sediment will also occur at higher concentrations in the organism (Ankley *et al.*, 1992; Lester and McIntosh, 1994) but there are exceptions. For example, PCBs from Kow of 5.0 to 7.0 have shown to have a greater tendency for uptake due to optimum lipophicity (Evans *et al.*, 1991; Boese *et al.*, 1997). Several authors also noted PCB uptake and desorption was not necessarily related to log Kow (Lake *et al.*,1996; Drouillard *et al.*, 1996; Wood *et al.*, 1997), as is the case for those individual PCB congeners that exhibit TCDD-like toxicity. These TCDD-like compounds include non-ortho-substituted congeners (e.g. PCB congeners 77, 126, 169), also referred to as co-planar compounds, can be preferentially accumulated from sediment (Lester and McIntosh, 1994; Lake *et al.*, 1995). The mono-ortho-substituted congeners tend to be less toxic (e.g. PCB congeners 105,118, 123, 114, 167) and exhibit varying degrees of uptake by organisms.

No-effect body burdens for certain PCB congeners have been reported: >100 μg/g in fathead minnows (Nebeker *et al.*, 1974), 71 μg/g in brook trout (Mauck *et al.*, 1978) and 54 μg/g in coho salmon (Mayer *et al.*, 1977). These tissue concentrations are well below those reached in the present study yet are at levels capable of inducing enzyme activity (Newsted *et al.*, 1995).

5.0 CONCLUSIONS

Laboratory sediment toxicity tests conducted in 1992 and 1996 on Lyons Creek surficial sediments found a range in toxicological effects among sites. Significant lethality (40% to 66% mortality) to the mayfly, *Hexagenia limbata* and the midge, *Chironomus tentans* occurred for station T5 and station 4 sediments in 1992. Several of the remaining test sediments had a negative impact on benthic growth, as compared to that attained for the reference control sediment. Both mayfly and midge growth was impaired to an equal extent with little difference in species sensitivity. None of the 1992 and 1996 sediments were found to be toxic to the fathead minnow, *Pimephales promelas*.

Sediment total polychlorinated biphenyl concentrations greater than 1.0 μg/g or 27 μg/g OC, were ordinarily associated with reduced mayfly and midge growth, of at least 50%. However, the chemical data did not adequately describe the lethality data. Mayfly and midge lethality occurred for those sediments with total PCB concentrations above 3.0 μg/g or 62 μg/g OC. The highest concentration of PCBs in sediment was 4-fold lower than the Provincial Sediment Quality Guideline Severe Effect Level concentration and was also substantially lower than the PCB effect-level concentrations reported in the literature. Additional chemical analysis for other organic compounds may help to determine the occurrence of other potential causative factors. It was noted that the presence of visible oil in the sample corresponded to organism lethality and may act as a marker for identifying sites of potential impact.

The 1992 and 1996 fathead minnow tests illustrated a high availability of total PCBs in laboratory-exposed fish after 21-days. The resulting total PCB whole-body tissue concentrations consistently exceeded the International Joint Commission Aquatic Life Guideline of 100 ng/g. A significant relationship was found between the total PCB concentration measured in the sediment and the concentration measured in the fathead minnows. This relationship suggests Lyons Creek surficial sediments are acting as a source of PCBs. The high availability of PCBs appears to be non-site-specific and relatively independent of the sediment total organic carbon content. The resulting biota-sediment accumulation factors, normalized for sediment percent organic carbon and fish percent lipid, averaged from 2.9 to 7.8.

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APPENDIX

TABLE A1. Analytical detection limits for nutrients, inorganic and organic contaminants in sediment an biota samples.

Parameter	Sediment (units: dry weight)
Nutrient :	
Loss on Ignition	1.0 mg/g
Total Organic Carbon	0.2 mg/g
Total Kjeldahl Nitrogen	0.025 mg/g
Total Phosphorus	-
Trace Metal :	
Arsenic	-
Cadmium	0.05 g/g
Chromium	1.0 g/g
Copper	0.5 g/g
Iron	200 g/g
Lead	1.25 g/g
Mercury	0.01 g/g
Nickel	0.2 g/g
Zinc	2.0 g/g
Organic:	
Naphthalene	20 ng/g
Acenapthylene	20 ng/g
Acenaphthene	20 ng/g
Fluorene	20 ng/g
Phenanthrene	20 ng/g
Anthracene	20 ng/g
Fluoranthene	20 ng/g
Pyrene	20 ng/g
Benzo[a]anthracene	20 ng/g
Chrysene	20 ng/g
Benzo[b]fluoranthene	20 ng/g
Benzo[k]fluoranthene	20 ng/g
Benzo[a]pyrene	40 ng/g
Benzo[g,h,i]perylene	40 ng/g
Dibenzo[a,h]anthracene	40 ng/g
Indeno[123-cd]pyrene	40 ng/g

Parameter	Sediment (units: dry weight)	Biota (unit: wet weight)
Organic :		
Total PCBs	20 ng/g	20 ng/g
Heptachlor	1 ng/g	1 ng/g
Aldrin	1 ng/g	1 ng/g
Mirex	5 ng/g	5 ng/g
a-BHC	1 ng/g	1 ng/g
b-BHC	1 ng/g	1 ng/g
g-BHC	1 ng/g	1 ng/g
a-Chlordane	2 ng/g	2 ng/g
g-Chlordane	2 ng/g	2 ng/g
Oxychlordane	2 ng/g	-
op-DDT	5 ng/g	5 ng/g
pp-DDD	5 ng/g	5 ng/g
pp-DDT	5 ng/g	5 ng/g
pp-DDE	5 ng/g	5 ng/g
Methoxychlor	5 ng/g	-
Heptachlor epoxide	1 ng/g	-
Endosulphan I	2 ng/g	-
Dieldrin	2 ng/g	-
Endrin	4 ng/g	-
Endosulphan II	4 ng/g	-
Endosulphan Sulphate	4 ng/g	-
Hexachloroethane	1 ng/g	1 ng/g
Hexachlorobutadiene	1 ng/g	1 ng/g
2,3,6-trichlorotoluene	1 ng/g	1 ng/g
2,4,5-trichlorotoluene	1 ng/g	1 ng/g
2,6,5-trichlorotoluene	1 ng/g	-
1,2,3-trichlorobenzene	2 ng/g	2 ng/g
1,2,4-trichlorobenzene	2 ng/g	2 ng/g
1,3,5-trichlorobenzene	2 ng/g	2 ng/g
1,2,3,4-tetrachlorobenz	1 ng/g	1 ng/g
1,2,3,5-tetrachlorobenz	1 ng/g	1 ng/g
1,2,4,5-tetrachlorobenz	1 ng/g	1 ng/g
Pentachlorobenzene	1 ng/g	1 ng/g
Hexachlorobenzene	1 ng/g	1 ng/g
Octachlorostyrene	1 ng/g	1 ng/g
Toxaphene	-	200 ng/g